Atmospheric Environment 43 (2009) 5908–5915

Contents lists available at ScienceDirect

Atmospheric Environment

journal homepage: www.elsevier.com/locate/atmosenv

The impact of wood stove technology upgrades on indoor residential air quality

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ARTICLE INFO

Article history:
Received 25 June 2009
Received in revised form
14 August 2009
Accepted 19 August 2009

Keywords:
Wood stove
Intervention
Particulate matter
Levoglucosan
Infiltration

ABSTRACT

Fine particulate matter ($PM_{2.5}$) air pollution has been linked to adverse health impacts, and combustion sources including residential wood-burning may play an important role in some regions. Recent evidence suggests that indoor air quality may improve in homes where older, non-certified wood stoves are exchanged for lower emissions EPA-certified alternatives. As part of a wood stove exchange program in northern British Columbia, Canada, we sampled outdoor and indoor air at 15 homes during 6-day sampling sessions both before and after non-certified wood stoves were exchanged. During each sampling session two consecutive 3-day $PM_{2.5}$ samples were collected onto Teflon filters, which were weighed and analyzed for the wood smoke tracer levoglucosan. Residential $PM_{2.5}$ infiltration efficiencies (F_{inf}) were estimated from continuous light scattering measurements made with nephelometers, and estimates of F_{inf} were used to calculate the outdoor- and indoor-generated contributions to indoor air. There was not a consistent relationship between stove technology and outdoor or indoor concentrations of $PM_{2.5}$ or levoglucosan. Mean F_{inf} estimates were low and similar during pre- and post-exchange periods (0.32 ± 0.17 and 0.33 ± 0.17, respectively). Indoor sources contributed the majority ($\sim$65%) of the indoor $PM_{2.5}$ concentrations, independent of stove technology, although low indoor-outdoor levoglucosan ratios (median ≤ 0.19) and low indoor $PM_{2.5}$-levoglucosan correlations ($r \leq 0.19$) suggested that wood smoke was not a major indoor $PM_{2.5}$ source in most of these homes. In summary, despite the potential for extensive wood stove exchange programs to reduce outdoor $PM_{2.5}$ concentrations in wood smoke-impacted communities, we did not find a consistent relationship between stove technology upgrades and indoor air quality improvements in homes where stoves were exchanged.

1. Introduction

Residential wood combustion (RWC) is a common heating method and a major source of air pollution in many locations in North America and northern Europe (Lepage and Boulton, 2000; Maykut et al., 2003; Glasius et al., 2006; Hellen et al., 2008; Karvosenoja et al., 2008). In Canada, RWC accounts for over 25% of fine particulate matter ($PM_{2.5}$) emissions (Lepage and Boulton, 2000). Despite potential air pollution impacts (Karvosenoja and Johansson, 2003), the use of wood is expected to increase due to rising costs of other fuels and the perception that wood is a "carbon neutral" fuel (Zezima, 2008; Richter et al., 2009).

RWC emissions are a complex mixture of various PM sizes, elemental and organic carbon, carbon monoxide, oxides of nitrogen, volatile organic compounds, and polycyclic aromatic hydrocarbons (Lighty et al., 2000; Naeher et al., 2007). Although in community settings it is not possible to isolate the health effects of wood smoke from other particle sources, epidemiologic studies of PM have reported associations with cardiopulmonary health effects in locations heavily impacted by RWC (Koenig et al., 1993, 2003; Schwartz et al., 1993; Lipsett et al., 1997; Norris et al., 1999, 2000; Slaughter et al., 2003; Trenga et al., 2006; Allen et al., 2008). For example, PM_{10} was associated with respiratory and cardiovascular hospital admissions in Christchurch, New Zealand, where approximately 90% of the wintertime PM_{10} is wood smoke (McGowan et al., 2002). A recent review summarized the wood smoke literature and found no persuasive evidence that wood smoke particles pose significantly less risk for respiratory disease than other major categories of similarly sized, combustion-derived particles (Naeher et al., 2007), though evidence regarding cardiovascular disease risks of wood smoke is still limited.

RWC emissions may be of particular population health concern because, relative to other sources (e.g., industry), there is high potential for exposure since emissions occur in residential areas (Glasius et al., 2006; Larson et al., 2007) and small quantities of pollutants may be emitted directly into the indoor environment

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doi:10.1016/j.atmosenv.2009.08.016

(Sexton et al., 1984). Ries et al. (2009) recently estimated the wood smoke intake fraction (the proportion of emissions inhaled by an exposed population) in Vancouver, British Columbia (BC) to be comparable to or slightly greater than typical urban intake fractions for vehicle emissions.

Older, non-certified wood stoves burn relatively inefficiently and have PM emission rates approximately 70% greater than EPA-certified wood stoves (Jaasma et al., 1993). Therefore, in an effort to reduce outdoor air pollution concentrations, non-certified stoves have been targeted for replacement in wood smoke-impacted communities in North America (Bergauff et al., 2009; EPA., 2009) and Europe (Nyrud et al., 2008). For example, a highly successful stove exchange program in and around Libby, Montana resulted in the replacement or surrender of nearly 1200 of an estimated 1300 non-certified wood stoves. Ambient $PM_{2.5}$ concentrations decreased 20% during the changeout period, while ambient concentrations of the wood smoke tracer levoglucosan decreased by 50% (Bergauff et al., 2009).

In addition to impacts on outdoor concentrations, stove exchanges could also improve indoor air quality by reducing indoor emissions (i.e., stove "leakage") and/or by reducing emissions outdoors leading to decreased indoor infiltration. Reductions in indoor concentrations would also reduce exposures since most people spend the vast majority of time indoors (Klepeis et al., 2001). A study in Libby found that average indoor $PM_{2.5}$ concentrations in 16 homes in which a non-certified stove was replaced with a certified model were reduced by 71%, from 51.2 to 15.0 $\mu g/m^3$, and average concentrations of levoglucosan were reduced by 45% (Ward and Noonan, 2008). However, because this study did not directly account for the infiltration of outdoor pollution, it was not possible to determine the relative impact of indoor emissions and infiltrated outdoor emissions on the observed decreases in indoor concentrations. To our knowledge, no other studies of wood stove upgrades and indoor air quality have been published.

We used a wood stove exchange program in northern BC as a platform from which to study the air pollution exposure impacts of upgrading from non-certified to certified wood stoves. Specifically, this study was conducted in two communities in the Bulkley Valley and Lakes District (BVLD) of BC, a region in which 7200 of 11,500 homes heat with wood, and 4200 (58%) of the wood-burning appliances are non-EPA-certified (BVLD Airshed Management Society., 2006). Our primary objective was to evaluate the impact of stove upgrades on indoor $PM_{2.5}$ concentrations in the homes where stoves are upgraded while adjusting for the influence of outdoor pollution.

2. Methods

2.1. Study design

This study was conducted from November, 2007 to April, 2008 in Smithers (population ~5300) and Telkwa (population ~1400), BC. Non-smoking households in which wood was the primary or secondary heating source were eligible to participate. Each study home was monitored during one 6-day monitoring period prior to the stove exchange and one 6-day monitoring period after the stove exchange. In addition to publicly available municipal and provincial rebates, study participants were provided with partial financial incentives to help offset the cost of the new stove. To allow study participants time to learn proper operation of their new stove, the post-exchange sampling period was conducted $\geq$ 4 weeks after the stove exchange took place. Field technicians collected data on home (e.g., square footage, volume, number of rooms, number of windows, presence of a major road within 100 m) and stove characteristics (e.g., type, brand, model, age, location). During each 6-day sampling period study participants kept a diary on which they recorded their activities (including home occupied/unoccupied, stove lit/unlit, stove door open/closed, wood added to stove, cooking, cleaning, and use of candles/incense) at hourly resolution. An attempt was made to schedule pre- and post-exchange sampling on the same days of the week to minimize the effect of weekday/weekend differences in stove use patterns.

2.2. Pollution measurements

During each 6-day monitoring period, $PM_{2.5}$ samples were collected onto Teflon filters during two consecutive 3-day samples using single-stage Harvard Impactors (Air Diagnostics and Engineering, Harrison, ME) and 10-lpm pumps (Leland Legacy, SKC Inc., Eighty Four, PA). To best capture residents' exposures the indoor sampling equipment was placed in the most frequently used living room, as far as possible from windows, doors, and ventilation sources. In homes where the wood stove was also in the living room, the monitoring equipment was placed as far as possible from the stove. Outdoor equipment was placed in a secure location near the home (in the yard or on a deck or patio), and not directly adjacent to trees, sheds, or other large objects. $PM_{2.5}$ mass concentrations were determined gravimetrically at the University of British Columbia School of Environmental Health (SOEH) laboratory. The limit of detection (LOD, calculated as 3 $\times$ SD of field blank mass divided by average sample volume) and precision (calculated as the SD of duplicate sample differences divided by the square root of 2) for $PM_{2.5}$ were 0.7 and 0.4 $\mu g/m^3$, respectively, and all $PM_{2.5}$ concentrations were above the LOD.

Finally, filter samples were analyzed for levoglucosan, a commonly used tracer of wood smoke particles (Naeher et al., 2007), by gas chromatography mass spectrometry (GC–MS) at the SOEH laboratory (Simpson et al., 2004). For levoglucosan the LOD and precision were 1.3 and 50.4 ng/m^3, respectively, and all measured concentrations were above the LOD.

The light scattering coefficient was measured at hourly resolution indoors and outdoors at study residences using portable integrating nephelometers (Radiance Research M903). The outdoor sampling line was heated to reduce the relative humidity (RH) since the relationship between $PM_{2.5}$ concentration and light scattering coefficient is nonlinear at high RH. Pairs of nephelometers used at each home were co-located for approximately 24 h following each 6-day sampling session to identify and adjust for any bias. Temperature and relative humidity were measured indoors using HOBO data loggers (Onset Computer Corp., Pocasset, MA).

In addition to measurements collected as part of this study, tapered element oscillating microbalance (TEOM) $PM_{2.5}$ data at centrally located pollution monitoring stations in both Smithers and Telkwa were obtained from the BC Ministry of Environment. Outdoor temperature data were obtained from the Environment Canada meteorological station in Smithers.

2.3. Data analysis

Partitioning the indoor $PM_{2.5}$ concentration into its indoor- and outdoor-generated components requires estimation of the $PM_{2.5}$ infiltration efficiency (F_{inf}). F_{inf}, defined as the fraction of the ambient concentration that penetrates indoors and remains suspended under steady-state conditions, depends on the particle penetration efficiency (P; unitless), the air exchange rate (a; h^{-1}), and the particle deposition rate (k; h^{-1}):

$$F_{inf} = \frac{Pa}{a+k} \quad (1)$$

Table 1
Characteristics of study homes and sampling periods.

Home	Community	Building type	Building age (yrs)	Square footage	Attached garage	Primary heating source	Pre-exchange sampling start	Pre-exchange sampling avg outdoor temp (°C)	Post-exchange sampling start	Post-exchange sampling avg outdoor temp (°C)	Old stove location	New stove location
1	Telkwa	Free-standing	30	2600	Yes	Wood	Nov 14	−1.4	Mar 5	1.1	Basement	Basement
2	Telkwa	Free-standing	60	1380	No	Gas	Nov 15	−2.3	Feb 7	−8.1	Kitchen	Living Room
3	Smithers	Trailer	16	910	No	Wood	Nov 21	−3.2	Feb 29	−0.6	Living Room	Living Room
4	Telkwa	Free-standing	34	1440	No	Wood	Nov 23	−3.9	Jan 11	−0.5	Living Room	Living Room
5	Telkwa	Free-standing	13	2700	No	Gas	Nov 29	−14.5	Jan 31	−12.4	Living Room	Living Room
6	Smithers	Free-standing	26	2600	No	Gas	Nov 30	−15.1	Feb 1	−2.2	Basement	Basement
7	Smithers	Free-standing	77	780	No	Wood	Dec 6	−9.0	Jan 4	−8.5	Living Room	Living Room
8	Smithers	Free-standing	30	2830	Yes	Wood	Dec 7	−10.9	Feb 8	−7.0	Basement	Basement
9	Smithers	Free-standing	30	2460	No	Oil	Dec 13	−3.4	Mar 7	1.5	Basement	Basement
10	Smithers	Free-standing	28	2900	No	Wood	Dec 14	−3.8	Feb 15	−3.3	Living Room	Living Room
11	Telkwa	Free-standing	23	2400	No	Wood	Jan 3	−6.9	Mar 27	−1.0	Basement	Basement
12	Telkwa	Free-standing	86	1900	No	Wood	Jan 10	−3.7	Mar 13	0.3	Living Room	Living Room
13	Smithers	Free-standing	39	2300	No	Wood	Jan 17	−8.8	Apr 4	2.1	Basement	Basement
15	Smithers	Trailer	14	1100	No	Wood	Jan 24	−16.8	Mar 14	0.3	Porch	Porch
17	Smithers	Free-standing	35	1750	Yes	Gas	Feb 26	0.6	Apr 1	2.8	Basement	Basement
Mean			36	2000				−6.9		−2.4		
Std Dev			22	73				5.4		4.5		
Median			30	2300				−3.9		−0.6		

The infiltration of outdoor $PM_{2.5}$ was assessed using the continuous indoor and outdoor light scattering data and a recursive model (RM), which has previously been described in detail (Allen et al., 2003, 2007). In summary, the RM states that the average indoor particle light scattering coefficient (b_{sp}) during hour t ($(b_{sp})_t^{in}$) is comprised of a fraction of the average outdoor particle scattering coefficient during the same hour ($(b_{sp})_t^{out}$), a fraction of the average indoor particle scattering coefficient remaining from the previous hour ($(b_{sp})_{t-1}^{in}$), and the scattering contribution from indoor sources (S_t^{in}) (Allen et al., 2003):

$$(b_{sp})_t^{in} = a_1(b_{sp})_t^{out} + a_2(b_{sp})_{t-1}^{in} + S_t^{in} \quad (2)$$

The influence of indoor sources is minimized by algorithms that identify hours influenced by indoor sources. Values during these hours are set to missing and are not included in the regression. F_{inf} can then be estimated by determining the coefficients a_1 and a_2 via multiple linear regression of eq (2) and using the following relationship:

$$F_{inf} = \frac{a_1}{1 - a_2} \quad (3)$$

Home specific estimates of F_{inf} were used, in combination with gravimetrically measured concentrations of outdoor (C_{out}) and indoor (C_{in}) $PM_{2.5}$, to estimate the indoor-generated (C_{in}^{ig}) and infiltrated indoor concentrations (C_{in}^{inf}):

$$C_{in}^{inf} = F_{inf} \times C_{out} \quad (4)$$

and

$$C_{in}^{ig} = C_{in} - C_{in}^{inf} \quad (5)$$

$PM_{2.5}$ concentrations measured at centrally located fixed monitoring sites indicated substantial temporal variation during the study period. Six-day average $PM_{2.5}$ concentrations measured by TEOM at the Smithers and Telkwa sites were highly correlated with concentrations at the Houston site ($r > 0.8$), approximately 40 km away, suggesting that the Smithers and Telkwa central sites capture airshed-wide temporal variability. Because pre- and post-exchange conditions were assessed at different times, we used Smithers and Telkwa central site TEOM data to adjust for temporal (between-day) variations in ambient $PM_{2.5}$ concentrations. Specifically, we used ratios of 6-day average home-based measurements to central site $PM_{2.5}$ measurements to isolate the impact of stove exchange from the effect of airshed-wide temporal changes in ambient concentration. Without this adjustment airshed-wide temporal concentration changes could incorrectly be attributed to the stove exchanges. Because central site (TEOM) and home (Harvard Impactor) measurements were made using different technologies the concentrations are not directly comparable.

3. Results and discussion

A total of 17 homes were enrolled, and pre- and post-changeout sampling was successfully completed in 15 homes, including 13 single family homes and 2 trailers (Table 1). None of the homes were located near major roads. The average time between pre- and post-exchange sampling was 66 days (range: 29–111 days). Outdoor temperature was lower on average during pre-exchange sampling (mean: −6.9 ± 5.4 °C) than during post-exchange sampling (mean: −2.4 ± 4.5 °C; $p < 0.01$) (Table 1). For all but one home, the post-exchange sampling period was warmer than the pre-exchange period, and for 5 homes the pre- and post-exchange sampling outdoor temperatures differed by more than 5 °C (Table 1). Indoor temperatures during pre-exchange (20.6 ± 2.8 °C) and post-exchange (21.3 ± 1.7 °C) sampling periods were similar.

Residents of 10 homes (homes 3–5, 7–11, 15, and 17 in Table 1) adequately completed a time-activity diary (i.e., provided responses during ≥80% of hours) during both pre- and post-exchange sampling (Table 2). During post-exchange sampling periods residents reported, on average, a lower frequency of stoves being lit

Table 2
Summary of stove use patterns during pre- and post-exchange sampling periods.

	Mean (min-max) percent of hours that the activity was reported		
	Stove burning	Stove door opened	Wood added to stove
Pre-Exchange	83.1% (38–100)	24.5% (8–43)	22.5% (8–40)
Post-Exchange	70.4% (24–100)	17.1% (10–34)	16.6% (10–34)

Note: Based on 10 homes with complete pre- and post-exchange time-activity data.

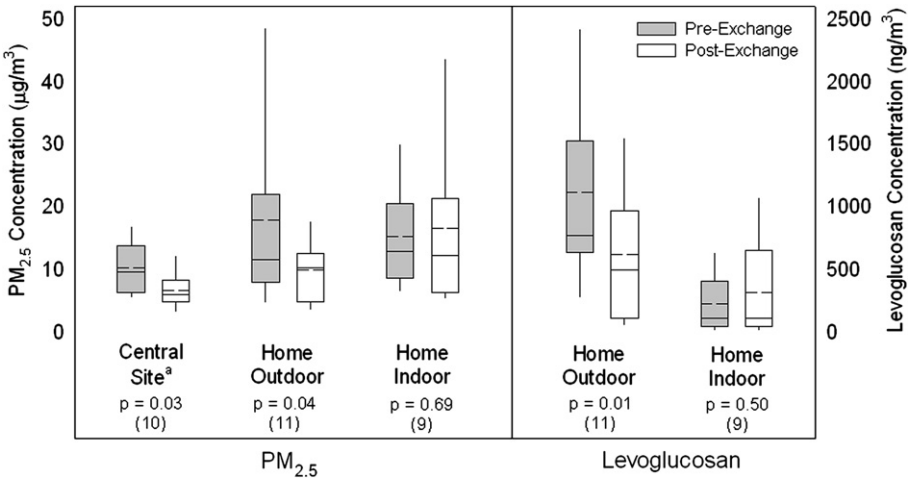


Fig. 1. Distributions of $PM_{2.5}$ and levoglucosan concentrations for pre-exchange and post-exchange sampling periods at 15 homes (whiskers represent 5th and 95th percentiles; outliers not shown).

(70.4% of post-exchange hours vs. 83.1% of pre-exchange hours, $p = 0.07$), opening the stove door (17.1% vs. 24.5%, $p < 0.05$), and adding wood to the stove (16.6% vs. 22.5%, $p < 0.05$). The average fraction of time that homes were occupied was similar during pre-exchange (85.5%) and post-exchange (81.9%) periods ($p = 0.21$).

Home outdoor $PM_{2.5}$ concentrations decreased after stove exchange at 11 of 15 homes, and the median change in outdoor $PM_{2.5}$ across all homes was -2.7 µg/m^3 ($p < 0.05$) (Fig. 1 and Table 3). Average outdoor concentrations of levoglucosan were also lower after stove exchange ($p < 0.01$), with a decrease at 11 homes and a median change of -406 ng/m^3 across all homes. Nine homes had a decrease in both $PM_{2.5}$ and levoglucosan outdoors. However, these changes were at least partly due to airshed-wide temporal changes in ambient concentrations. Central site $PM_{2.5}$ concentrations were generally lower during post-exchange sampling periods (median change: -3.4 µg/m^3; $p < 0.05$) (Fig. 1 and Table 3). After accounting for these temporal trends at the central site, there was not a consistent effect of stove technology on outdoor $PM_{2.5}$ concentrations, with outdoor/central concentration ratios increasing at 7 homes and decreasing at 8 homes following stove exchange (Fig. 2a).

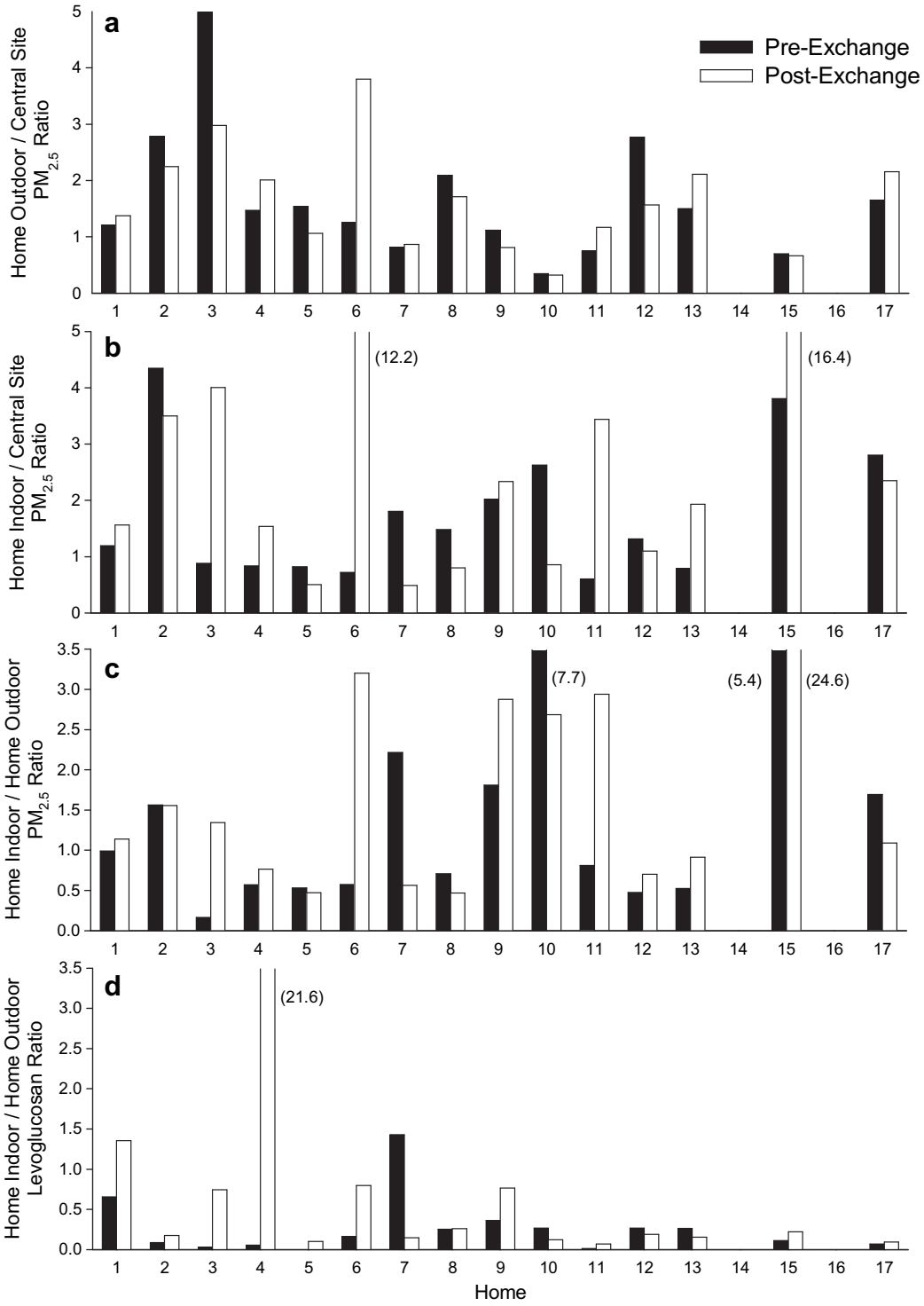
There was not a consistent effect of wood stove exchanges on indoor concentrations or indoor-outdoor ratios of $PM_{2.5}$ or levoglucosan. Average indoor $PM_{2.5}$ concentrations were similar for pre-exchange (median: 12.8 µg/m^3) and post-exchange (median: 12.2 µg/m^3) sampling sessions (Fig. 1), and nine homes had a decrease in indoor concentration following stove exchange (Table 3). There was also not a clear pattern in indoor $PM_{2.5}$ concentrations after normalizing to central site $PM_{2.5}$ to account for temporal changes in ambient concentration; the indoor/central site $PM_{2.5}$ ratio decreased in 7 homes and increased in 8 homes after stove exchange (Fig. 2b). Indoor-outdoor $PM_{2.5}$ ratios were variable, with ratios > 1 in several homes indicating substantial indoor $PM_{2.5}$ sources (Fig. 2c). Following stove exchange the indoor-outdoor $PM_{2.5}$ ratio decreased in 6 homes and increased in 9 homes (Fig. 2c). For levoglucosan, pre-exchange (median: 113 ng/m^3) and post-exchange (median: 109 ng/m^3) concentrations were also similar (Fig. 1). Indoor concentrations of levoglucosan decreased in 9 homes after stove exchange (Table 3), while indoor-outdoor ratios decreased in only 4 homes (Fig. 2d). Six homes had decreases in both $PM_{2.5}$ and levoglucosan concentrations indoors (Table 3). There were no clear relationships between stove exchange effects and housing, stove, or meteorological factors such as home age, home size, primary heating source (wood or other), stove location (living room or other), or outdoor temperature. Unfortunately, pre-exchange stove age was unknown at 5 homes, so we were unable to evaluate the role of stove age on the exchange's impact.

Valid indoor and outdoor nephelometer measurements from which to estimate F_{inf} were available for 9 homes (Table 4). The F_{inf} values in these homes were low during both pre-exchange (mean ± SD: 0.32 ± 0.17) and post-exchange sampling (0.33 ± 0.11). Given the relationship between F_{inf} and a, one would expect lower F_{inf} in colder regions due to energy efficiency requirements (Murray and Burmaster, 1995). Our F_{inf} estimates are similar to previous wintertime estimates in this region of BC (Barn et al., 2008), but lower than recent estimates in settings with milder winters (Allen et al., 2003; Wallace and Williams, 2005; Hystad et al., 2009).

Table 3
Changes in $PM_{2.5}$ and levoglucosan concentrations between pre- and post-exchange sampling periods. Decreases following stove exchange are in bold text.

Home	$PM_{2.5}$ changes			Levoglucosan changes	
	Central[a] (µg/m^3)	Outdoor (µg/m^3)	Indoor (µg/m^3)	Outdoor (ng/m^3)	Indoor (ng/m^3)
1	0.9	2.4	4.1	1	463
2	0.2	**−2.7**	**−4.3**	129	131
3	**−6.5**	**−52.8**	19.1	**−1767**	1211
4	**−15.0**	**−19.3**	**−9.0**	**−1631**	557
5	4.4	1.0	**−0.2**	325	96
6	**−7.2**	**−2.7**	23.3	211	607
7	**−3.4**	**−2.3**	**−18.0**	**−154**	**−858**
8	**−6.5**	**−16.3**	**−14.6**	**−589**	**−143**
9	**−0.3**	**−2.2**	1.3	**−406**	**−63**
10	6.4	1.9	**−5.7**	**−44**	**−24**
11	**−10.0**	**−5.9**	4.7	**−658**	**−4**
12	**−0.8**	**−9.1**	**−2.3**	**−961**	**−298**
13	**−9.7**	**−11.6**	**−2.1**	**−1074**	**−330**
15	**−5.8**	**−4.2**	25.1	**−280**	**−23**
17	0.4	3.2	**−1.3**	**−594**	**−30**
Median Change	−3.4	−2.7	−1.3	−406	−23
Paired t-test p-value	0.03	0.04	0.69	<0.01	0.50
# of Homes with Decrease	10	11	9	11	9

[a] Concentrations measured at the closest centrally located monitoring site during the home monitoring session with a tapered element oscillating microbalance (TEOM). Because different sampling technologies were used at central (TEOM) and home (Harvard Impactor) sites the two cannot be directly compared; central site data are used only for identifying temporal trends in airshed-scale $PM_{2.5}$ concentrations.



Note: Post-exchange sampling was not completed in homes 14 and 16.

Fig. 2. Concentration ratios of (a) home outdoor/central site $PM_{2.5}$, (b) home indoor/central site $PM_{2.5}$, (c) home indoor/home outdoor $PM_{2.5}$, and (d) home indoor/home outdoor levoglucosan. (Note: home/central site ratios > 5 and indoor/outdoor ratios > 3.5 are shown in parentheses).

We hypothesized that wood stove exchanges might improve indoor air quality in two ways. First, reduced outdoor emissions may result in decreased indoor concentrations, particularly in homes with high infiltration efficiencies. On average, the outdoor-generated $PM_{2.5}$ concentration indoors decreased following stove exchange (median change: $-1.6\ \mu g/m^3$; $p = 0.05$), but this decrease was eliminated after adjusting for temporal changes in ambient $PM_{2.5}$ concentration at the central monitoring sites. Alternatively, certified wood stoves might impact indoor air quality through reduced leakage directly into the indoor environment. Although indoor sources of $PM_{2.5}$ were responsible for approximately 65% of the indoor $PM_{2.5}$ concentration (consistent with the relatively high indoor-outdoor $PM_{2.5}$ ratios in Fig. 2c), on average, both before and after stove exchange, we found that the stove exchanges did

Table 4
Estimates of infiltration efficiency and indoor- and outdoor-generated indoor $PM_{2.5}$ concentrations at 10 homes with valid indoor and outdoor nephelometer data. ($PM_{2.5}$ concentrations are in $\mu g/m^3$) Decreases following stove exchange are in bold text.

Home	Pre-exchange				Post-exchange				Change	
	F_{inf}	Outdoor-generated $PM_{2.5}$	Indoor-generated $PM_{2.5}$	% Indoor-generated	F_{inf}	Outdoor-generated $PM_{2.5}$	Indoor-generated $PM_{2.5}$	% Indoor-generated	Outdoor-generated $PM_{2.5}$	Indoor-generated $PM_{2.5}$
1	0.37	3.4	5.5	62.2	0.12	1.3	11.6	89.6	**−2.1**	6.1
6	0.23	2.8	4.3	60.7	0.40	3.8	26.5	87.4	1.0	22.2
7	0.70	7.1	15.3	68.4	0.36	2.9	1.6	35.5	**−4.2**	**−13.7**
8	0.28	8.2	12.3	60.2	0.36	4.5	1.4	23.6	**−3.7**	**−10.9**
11	0.17	1.8	6.6	78.9	0.51	2.3	10.9	82.8	0.5	4.3
12	0.41	7.4	1.3	14.5	0.38	3.5	2.9	45.6	**−3.9**	1.6
13	0.21	4.6	6.9	59.9	0.29	3.0	6.4	68.1	**−1.6**	**−0.5**
15	0.16	1.1	35.2	97.0	0.31	0.8	60.6	98.7	**−0.3**	25.4
17	0.32	2.6	11.0	81.2	0.24	2.7	9.6	78.0	0.1	**−1.4**
Mean	0.32	4.3	10.9	64.8	0.33	2.8	14.6	67.7	−1.6	3.7
Std Dev	0.17	2.6	10.1	22.7	0.11	1.2	18.9	26.5	2.0	13.2
Median	0.28	3.4	6.9	62.2	0.36	2.9	9.6	78.0	−1.6	1.6

not consistently impact indoor-generated $PM_{2.5}$ concentrations (Table 4).

Outdoors, there was a strong relationship between $PM_{2.5}$ and levoglucosan, suggesting, as expected, that wood smoke is a major contributor to the variability in outdoor $PM_{2.5}$ in these communities (Fig. 3). In contrast, indoor $PM_{2.5}$ and levoglucosan were poorly correlated, which provides indirect evidence that the indoor $PM_{2.5}$ in these homes was primarily from other (non-wood smoke) sources. Generally low indoor-outdoor levoglucosan ratios during pre-exchange (median ratio: 0.16) and post-exchange (median ratio: 0.19) sampling also indicated a minimal contribution of indoor wood smoke sources in most homes, though indoor/outdoor levoglucosan ratios >1 during one pre-exchange and two post-exchange sessions indicate indoor wood smoke emissions in select cases (Fig. 2d). Participants' activity logs suggested that while at some homes opening the stove door was associated with large "peaks" in the hourly indoor nephelometer data, cooking also frequently corresponded to such peaks in several homes.

The literature on indoor wood smoke emissions is limited. In a 1982 study in Vermont, Sexton et al. (1984) measured indoor and outdoor respirable particle (RSP) concentrations at 19 non-smoking homes with and 5 non-smoking homes without wood-burning appliances. The average indoor-outdoor ratio was 1.6, indicative of indoor pollution sources, and wood-burning homes had higher average indoor RSP concentrations (24 $\mu g/m^3$) than homes not

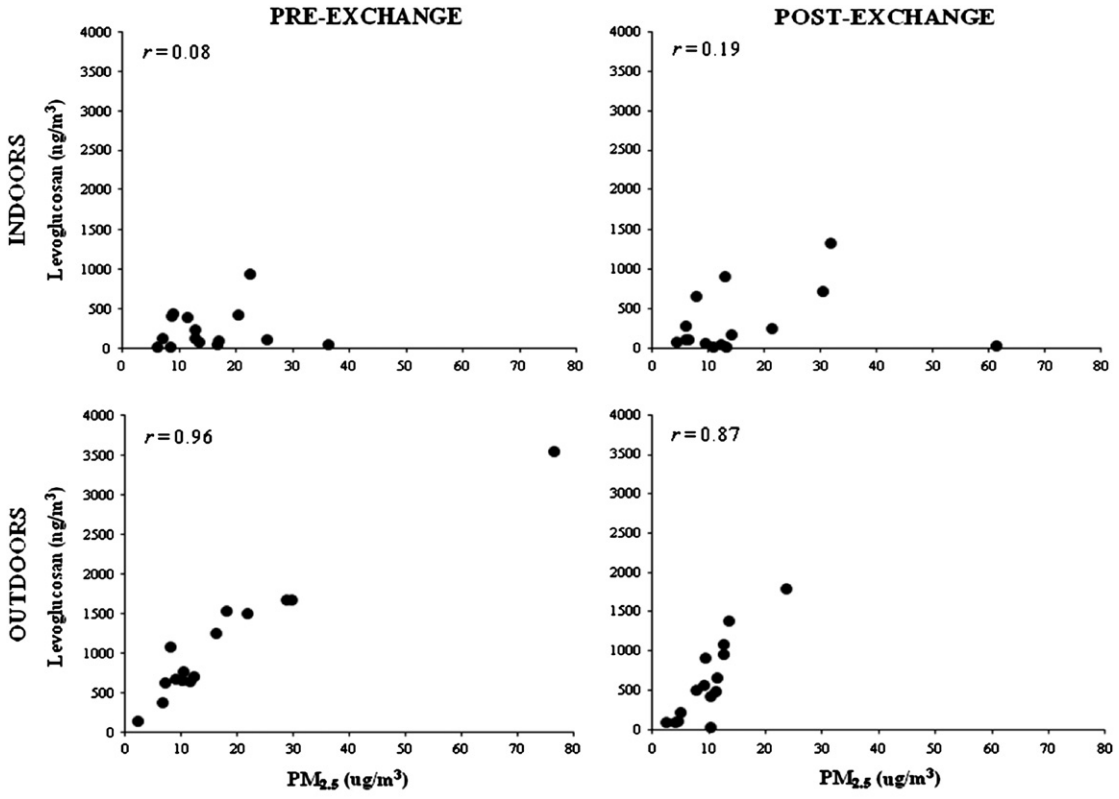


Fig. 3. Relationships between $PM_{2.5}$ and levoglucosan concentrations by location and stove exchange status.

burning wood (18 $\mu g/m^3$). A more recent study in Sweden reported that indoor levels of polycyclic aromatic hydrocarbons in 13 homes with wood-burning appliances were 3–5 times higher than concentrations in non-wood-burning homes (Gustafson et al., 2008). Unfortunately, since infiltration was not measured in these studies it is not possible to directly assess the relative impact of indoor and infiltrated emissions.

Our finding that stove exchanges did not consistently improve indoor air quality differs substantially from the Libby study's results (Ward and Noonan, 2008). It is important to note that unlike in Libby where nearly all non-certified stoves were exchanged, the BC exchange program is in the early stages and this study was conducted while a substantial number of non-certified stoves in the region were still in operation. In addition, the pre-exchange indoor levels were substantially lower than in Libby homes ($PM_{2.5}$ medians: 12.8 vs. 34.5 $\mu g/m^3$; levoglucosan medians: 113 vs. 652 ng/m^3), perhaps due to differences in pre-exchange stove installation quality and/or stove operations between the two studies. The air quality improvements reported in Libby may have been due, at least in part, to improvements in stove installation and/or operating procedures following stove exchange. It is also possible that our relatively short sampling durations (6 days before and after stove exchange) did not adequately capture long-term conditions in these homes, though the Libby study reported stove exchange effects using only 24-hr measurements to assess pre- and post-exchange conditions.

4. Conclusion

Changing from non-certified to certified wood stoves did not consistently reduce indoor concentrations of $PM_{2.5}$ or the wood smoke tracer levoglucosan in 15 BC homes. Although outdoor $PM_{2.5}$ concentrations were lower on average following stove exchange, these differences were largely the result of airshed-wide temporal changes in ambient $PM_{2.5}$ concentrations in these communities. Infiltration efficiencies were relatively low, and indoor sources of $PM_{2.5}$ contributed approximately $^2/_3$ of the total indoor concentration regardless of stove technology. Indirect evidence suggested that the indoor-generated $PM_{2.5}$ in these homes was mostly from sources other than RWC. If stove technology upgrades improve air quality primarily through reduced outdoor emissions, then the potential exposure reduction of community stove exchange programs may largely depend on the magnitude of residential infiltration efficiencies. Substantial wood smoke exposure reductions may only be possible if a large proportion of a community's non-certified stoves are replaced or removed.

Acknowledgments

We are very grateful to the study participants, Ingrid Granlin and Jeremy Roscoe for conducting the field work for this study, and Barbara Karlen for assistance with data preparation. We also appreciate the advice and support provided by Ben Weinstein and Colin MacLeod of the BC Ministry of Environment office in Smithers. This work was funded by the British Columbia Ministry of Environment, the BC Lung Association, Health Canada, and the BC Environmental and Occupational Health Research Network.

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